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Supporting non-target arthropods in agroecosystems: Modelling effects of insecticides and landscape structure on carabids in agricultural landscapes

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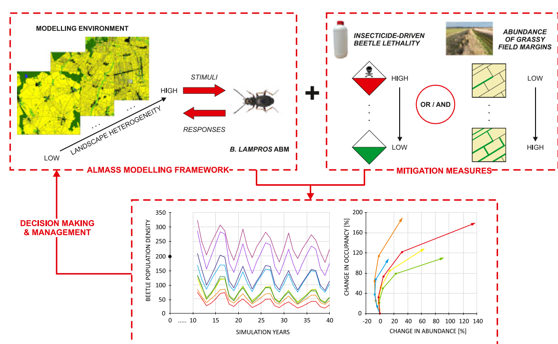
HIGHLIGHTS

- We simulated relative effects of agricultural intensification on beetle populations.
- Beetle population dynamics were assessed along a gradient of landscape heterogeneity.
- Reducing the insecticide-driven lethality was a major mitigation measure.
- Wide field margins supported beetles even with moderate use of insecticides.
- Effectiveness of mitigation measures strongly depends on landscape heterogeneity.

GRAPHICAL ABSTRACT



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ABSTRACT

Intensification of agricultural practices is one of the most important drivers of the dramatic decline of arthropod species. We do not know, however, the relative contribution to decline of different anthropogenic stressors that are part of this process. We used high-resolution dynamic landscape models and advanced spatially-explicit population modelling to estimate the relative importance of insecticide use and landscape structure for population dynamics of a widespread carabid beetle *Bembidion lampros*. The effects of in-crop mitigation measures through the application of insecticides with reduced lethality, and off-crop mitigation measures by increasing abundance of grassy field margins, were evaluated for the beetle along the gradient of landscape heterogeneity. Reducing the insecticide-driven lethality (from 90 to 10%) had larger positive impacts on beetle density and occupancy than increasing the abundance of field margins in a landscape. The effects of increasing field margins depended on their width and overall abundance in the landscape, but only field margins 4 m wide, applied to at least 40% of fields, resulted in an increase in beetle population density comparable to the scenario with the smallest reduction of insecticide-driven lethality we considered. Our findings suggest the importance of field margins rather as a supporting not stand-alone mitigation measure, as they generally improved effects of reduction of insecticide-driven lethality. Therefore, adding sufficiently broad off-field habitats should help to maintain viable beetle populations in agricultural landscapes even with moderate use of insecticides. In general, the less persistent the insecticides are in the environment, the larger positive impacts of applied mitigation

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measures on beetle populations were found. We also showed that the effectiveness of applied mitigation measures strongly depends on landscape and farmland heterogeneity. Thus, to achieve the same management or mitigation target in different landscapes might require different strategies.

1. Introduction

Over the last few decades, a dramatic decrease in abundance, diversity and biomass of arthropod species, including carabid beetles, has been observed in Europe (Brooks et al., 2012; Geiger et al., 2010; Hallmann et al., 2017, 2020; Homburg et al., 2019; Johan Kotze et al., 2011; Seibold et al., 2019; Van Noordwijk et al., 2017) and other regions of the world (Sánchez-Bayo and Wyckhuys, 2019; Wagner, 2020). Among the main drivers of these declines are land-use changes associated with the transition from natural to agricultural lands and intensification of agricultural practices including land consolidation and increase in pollution by pesticides and fertilizers (Brooks et al., 2012; Cardoso et al., 2020; Sánchez-Bayo and Wyckhuys, 2019; Wagner, 2020).

The agricultural landscapes of Europe have changed substantially in the last few decades, generally shifting from being dominated by small-scale family-run farms with diverse cropping system and a dense network of field margins and/or hedgerows towards more large-scale cultivation of uniform crops, extending sometimes for dozens of square kilometres. Indeed, the most evident and policy-relevant structural developments in European Union (EU) agriculture are reflected in consolidation of agricultural land. Although the total area of land used for agricultural production in EU remained broadly unchanged (+0.2%) between 2005 and 2016, the number of farms declined by almost 30%, and in consequence the average farm size in EU increased from 11.9 to 16.6 ha (Eurostat, 2018). The largest declines were observed in number of smallest farms (−38% for farms <2 ha and − 28% for farms <5 ha), while the increase was observed only in the group of largest holdings (+18% for farms ≥100 ha; Eurostat, 2018). As farm sizes grow, farms tend to specialise in cereals and livestock, moving away from permanent crops and mixed farming. At the same time, according to Eurostat, the consumption of pesticides is still increasing in many European countries, including the biggest consumers; e.g., between 2011 and 2018 insecticide sales increased by 25% in Romania, 37% in Germany, 79% in Poland and by more than 150% in France.

Increasing pressures of agricultural intensification on biodiversity are related to processes acting at various spatial and temporal scales (Benton et al., 2003), from increased application of pesticides and chemical fertilizers on individual fields to reduction in spatiotemporal heterogeneity of land use (simplification of crop rotations or loss in semi-natural habitats), at farm and landscape scales. In intensively managed agricultural landscapes, the ongoing loss and fragmentation of semi-natural habitats (Cousins et al., 2015; Ridding et al., 2020) has led to their reduction to remnants mainly in a form of linear features, such as field margins, hedges, and ditches or grassy strips along waterlines or roads. Nevertheless, these narrow, extensively managed, linear semi-natural habitats can play a key role in enhancing NTA species by locally reducing negative impacts of pesticides (De Snoo and De Wit, 1998; Reichenberger et al., 2007), serving as connecting corridors and source habitats for recolonization of fields (Cole et al., 2012; Hof and Bright, 2010), and providing structural attributes and vegetation that offer shelter, food resources and overwintering refugia (Geiger et al., 2009; MacLeod et al., 2004; Pfiffner and Luka, 2000). However, with progressive land consolidation, increasing field sizes and decreasing field boundaries density, the abundance of linear semi-natural habitats in agroecosystems is declining together with multiple ecosystem services provided by NTA (Mkenda et al., 2019).

Although there is a general consensus that intensification of agriculture is the main cause of the dramatic decline in NTA populations, the relative effects of different stressors generated by this process are unknown; i.e. whether the extensive use of pesticides, decreasing landscape heterogeneity, large-scale monocultures, or/and the lack of suitable habitats serving as refuges for NTAs plays the most important role. Considering the magnitude of NTA decline in Europe and elsewhere, and their crucial importance for ecosystem functioning and economy, resolving this problem is urgent, but cannot be done without detailed understanding of the mechanisms driving the NTA decline. These mechanisms, however, should be analyzed at various spatial scales simultaneously, taking into account management of local fields but also the context of agriculture and land-use at the landscape level (Seibold et al., 2019).

In recent years it became apparent that the protection goals and environmental risk assessment (ERA) schemes must be shifted from individual toxicity testing and single compound regulation towards a population-level approach and ecosystem services (Forbes et al., 2009; Landis, 2003; Nienstedt et al., 2012). This is because in real agricultural landscapes the anthropogenic stressors mentioned above co-vary, and their effects on NTAs depend on particular landscape configuration and complex spatial and temporal population dynamics, including 'action at a distance' or 'source-sink' phenomenon (Focks et al., 2014; Topping et al., 2014; Topping and Lagisz, 2012; Topping et al., 2020). Therefore, using either very simplified representations of the landscape structure or small sections of landscape (local / field scale of analysis) to predict population-level responses to stressors can induce heavy and unpredictable bias in the assessments (Holland et al., 2007; Topping et al., 2014). Integrating spatially-explicit population models with dynamic landscape modelling has been suggested as a solution allowing to capture long-term population responses to both spatio-temporal changes in habitat conditions and intensities of stressors (Dalkvist et al., 2013; Topping et al., 2015, 2016).

The objective of our research was to estimate the relative importance of insecticide use and landscape structure (e.g., size of fields, amount of semi-natural habitats and field margins) for population dynamics of a model NTA species *Bembidion lampros*, the representative of a large, diverse and important group of ecosystem service providers – the carabid beetles. Carabids, being natural pest enemies (Kromp, 1999; Symondson et al., 2002) and weed seeds regulators (Bohan et al., 2011; Kulkarni et al., 2015) in agricultural landscapes, are especially sensitive to human-altered abiotic conditions, and therefore can be treated as indicators of 'ecosystem health' (Koivula, 2011). By combining the detailed knowledge of the species biology, high-resolution dynamic landscape models, and spatially-explicit population modelling, we tested which management strategies would most effectively support *B. lampros* populations in agro-ecosystems. In particular, we compared the impact of (1) in-crop mitigation measures related to insecticide treatments, i.e., using less toxic insecticides or at lower doses, (2) off-crop mitigation measures related to increasing abundance of field margins in a landscape, and (3) a combination of both, on abundance and occupancy of *B. lampros* in selected agricultural landscapes in Poland.

The Animal, Landscape and Man Simulation System (ALMaSS) (Topping et al., 2003) used in this study has been already applied to evaluate the impact of changing insecticide toxicity and landscape structure on population density and the distribution of beetles in exemplary landscapes in Denmark (Topping et al., 2015, 2019) and in the Netherlands (Ziółkowska and Topping, 2019). However, findings obtained in these two countries may not be fully applicable across

Europe, as each country is probably unique in its combination of landscape structure and management. In contrast to Denmark and the Netherlands, the farm sizes in Poland are on average much smaller (according to Eurostat (2018) the mean size of agricultural holding in Poland in 2016 was 10.2 ha, compared to 32.3 ha in the Netherlands and 74.6 ha in Denmark), and the farmland environment is generally less extensive with a domination of mixed and hobby production. In addition, in Poland within a relatively small area, hugely different landscapes can be found, ranging from traditional small-scale farming to large-scale agriculture. Therefore, Polish agroecosystems offer a unique opportunity to study effects of drastically different agricultural landscape managements within the same climatic conditions. We used this uniqueness of Polish agroecosystems to evaluate how the effects of different mitigation measures can differ along the gradient of landscape heterogeneity for population dynamics of *B. lampros*.

2. Methods

The simulation system used, ALMASS, is an open-source project hosted on GitLab (https://gitlab.com/ALMaSS/) with online documentation (https://projects.au.dk/almass/documentation/) using the ODdox (Overview Design doxygen) protocol (Topping et al., 2010). ALMaSS integrates agent-based models of selected species with detailed description of an environment (landscape) from which modeled individuals obtain information necessary to simulate their behavior.

The methods comprise three sections, describing (i) the design of a landscape model in ALMaSS and its parametrization for the Polish landscapes; (ii) the species model used and its calibration to the Polish climatic conditions; and (iii) scenarios design and analysis for evaluating mitigation strategies and landscape effects.

2.1. Parametrization of the ALMaSS landscape component for the Polish landscapes

2.1.1. Dynamic landscape model in ALMaSS

The landscape component in ALMaSS needs to provide a realistic environment to support agent-based simulation of species, it includes both spatial and temporal aspects of landscape heterogeneity. The spatial landscape component consists of a detailed land cover raster map with complete coverage and spatial resolution of 1 m^2, and can support modelling of species with widely differing ecologies, e.g. ground beetles to skylarks, and deer. Each pixel in the raster map is classified in accordance with its landscape element type (e.g., natural, build-up, cultivable), including detailed structures as hedgerows, individual trees or road verges. Cultivable areas are described in more detail, through delineation of agricultural parcels (fields) and grouping of these into farm units of different types (e.g., conventional cattle, pig or arable farms). Landscape elements are also characterized by other environmental attributes, such as main soil type.

Temporal landscape component needs to account for the high dynamics of agricultural landscapes, both within-year and pluriannual. In ALMaSS, all vegetation types, including crops, have associated growth models describing daily changes in vegetation height, green and total biomass in response to weather conditions. In addition, all agriculturally managed vegetation types respond to individually tailored management plans consisting of logical and ordered combinations of farm activities (related to soil cultivation, application of fertilizers and pesticides), time windows within which they may occur with the probabilities of carrying out the activity and conditions under which the activity may take place. These conditions may relate to weather, crop growth, soil type or previous farming activities. Pluriannual temporal heterogeneity relates to crop rotations, defined separately for each farm type. This approach gives a highly realistic, daily updated, dynamic landscape with vegetation growing in response to the weather, and the pattern of farming activities related to each specific crop, farm, and field (Topping et al., 2016).

2.1.2. 'Capturing' the Polish agricultural system

All details related to the generation of landscape model for Poland are described in Appendix A, with an overview given here. This approach broadly followed Topping et al. (2016) but required specific Polish conditions and datasets to be taken into account.

Four main types of data were gathered and processed to obtain the landscape inputs necessary for ALMaSS:

1. *Land cover / land use information*. The data were derived from the National Database of Topographic Objects (BDOT) being a part of nationwide system of collecting and sharing topographic data in the national spatial data infrastructure (SDI). At present, BDOT provides the most precise information level on land cover and land use in Poland, and is appropriate for maps in the scale 1 : 10 000. BDOT includes 10 thematic areas: administrative units, network of roads and railways, buildings and installations, land cover, land use, the network of watercourses, protected areas, geodetic networks and objects, the network of public utility lines, and points of address. We used 27 classes of objects from different BDOT thematic areas to map 64 different layers of information (see Fig. A1 and Table A1 in Appendix A). In addition, information on managed permanent grasslands was obtained from the Polish Agency of Restructuring and Modernization of Agriculture (ARMA) to update and divide grassland objects mapped within BDOT land cover thematic area (PTTR01 objects) into managed and unmanaged (semi-natural) permanent grasslands.
   Individual layers of land use / land cover information together with information on agricultural fields (details provided below) were then combined into a single raster landscape map in a step-by-step process. The use of layers from different data sources resulted in inconsistencies related to spatial alignment of features (overlaps or gaps between features). In addition, some objects were represented as points or lines and therefore as dimensionless had to be first preprocessed in order to change them into two-dimensional ones. This process also contributed to the number of inconsistencies in the map with combined layers. Therefore, a special step-by-step procedure was applied to intelligently correct these inconsistencies to obtain a landscape raster map with no gaps in information (see details in Appendix A).
2. *The Land Parcel Identification System (LPIS)*. LPIS records information on all agriculturally managed reference parcels (geographically delimited areas with unique identification codes) in the EU Member States and serves as a controlling mechanism under the Common Agricultural Policy (CAP). In Poland, LPIS managed by the ARMA, is based on the national land and building cadaster and therefore the cadastral parcel serves as the reference parcel for the LPIS. From LPIS we used information for 2018, on type of crops cultivated in reference parcels (from the register of direct payments), and ID numbers of agricultural holdings enabling the grouping of individual reference parcels into farm units. We also used LPIS to divide the cadastral parcels of cropland mapped within the BDOT land cover thematic area (PTTR02 objects) into agricultural fields. If, according to the register of crops from LPIS, more than one crop was cultivated within one reference parcel, we divided the parcel artificially based on crop area, along the polygon's orientation axis. Similarly, if two or more directly neighboring reference parcels belonged to the same agricultural holding and were managed together (i.e., were covered by the same crop), we considered them as one agricultural field.
3. *Animal Identification and Registration System (AIRS)*. AIRS is a database of tagged farm animals, i.e. cattle, ovine, caprine and porcine animals, managed by the ARMA. We used database on farm animals together with the register of crops to classify farm units into general farm types: vegetable, potato, beet, cattle, pig, mixed stock, arable and hobby farms (see details in Appendix A). Based on proportions of crops cultivated by farms of different types, crop rotation schemes were generated for each farm type individually. Only crops with

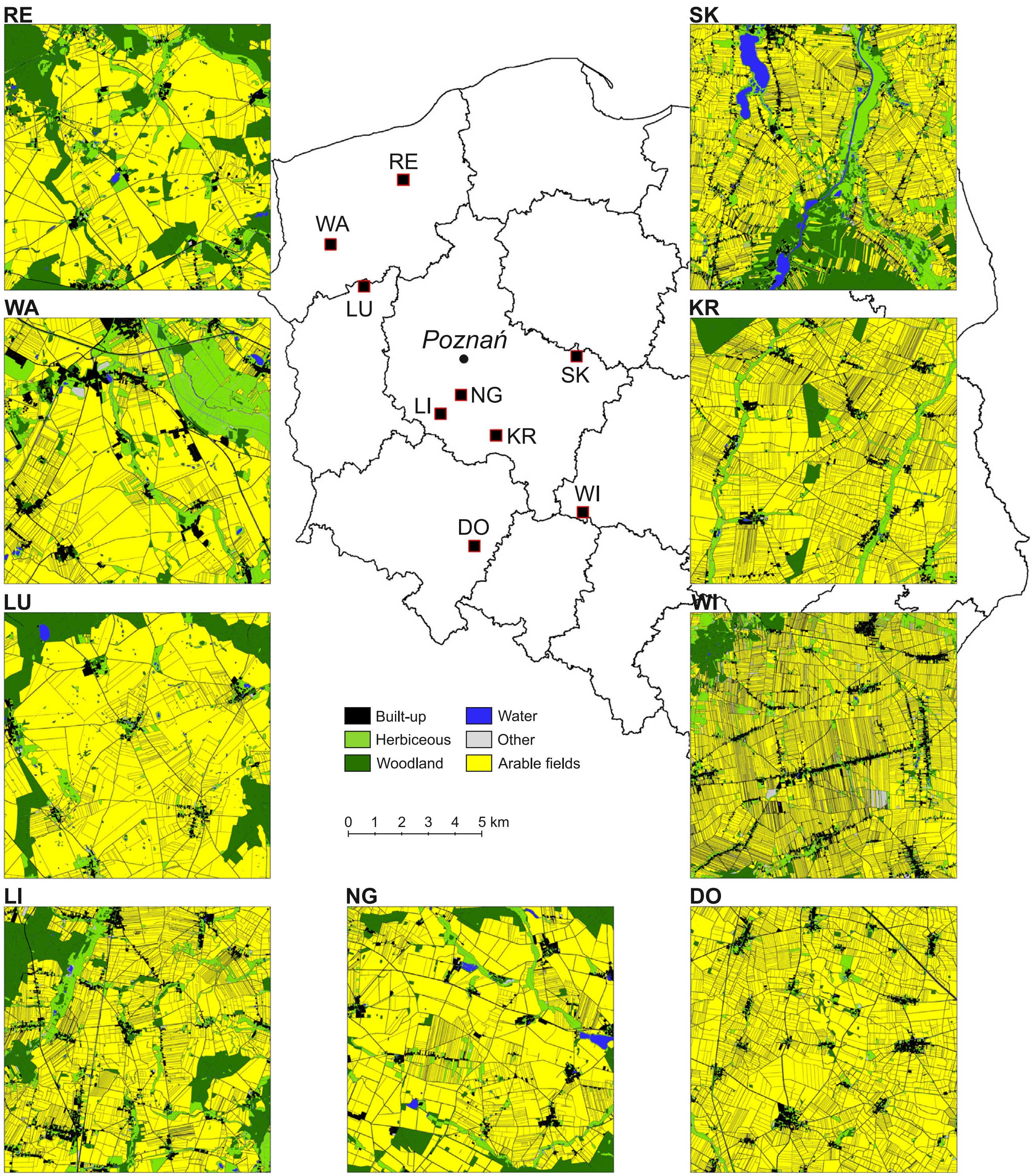

Fig. 1. Landscape structure in nine 10 × 10 km study areas. Background map shows location of the study areas in the Polish provinces. Key denotes general classes of landscape elements visible on land-use maps for study areas. Although spatial resolution is 1 m^2, not all features are visible (including narrow grassy features that are important habitats for *B. lampros*) due to the scale used. For the same reason it was impossible to show all the landscape elements classes used in detailed ALMaSS landscape models of the study areas.

more than 1% share of the area of a farm type were considered. It was assumed that the rotation could be represented by 100 crops (1 crop for each 1%). The order of crops followed typical agronomic practices and issues such as late harvest leading to impossible sowing conditions were controlled by the built-in ALMaSS farm code. The result was a pattern of changing crops on a field that matches the overall crop distribution pattern for that farm type precisely over 100 seasons. If a specific crop, e.g., maize for silage, occurs 13 times out of 100 in the rotation, it will on average occur on 13% of all fields covered by that rotation at any point in time.

4. *Up-to-date crop management plans*. These plans for dominant Polish field crops (winter wheat, winter barley, winter triticale, winter rape, winter rye, spring wheat, spring barley, potatoes, maize, carrot, cabbage, beans, beet, and fodder lucerne) including time windows and probabilities of occurrence of main soil cultivation practices, as well as fertilizer and pesticide applications (together with information on

product used and its dosage) were obtained from the farmer advisors of the Agricultural Advisory Centre, regional office in Wielkopolska province, and/or were designed based on methods of integrated plant production published by the Main Inspectorate of Plant Health and Seed Production (https://piorin.gov.pl/publikacje/metodyki-ip/).

All handling and analysis of spatial data were done using Python 2.7 and the Python library arcpy to access ArcGIS features or directly in ArcGIS 10.4 (ESRI, 2016). The entire process of producing a Polish landscape model for ALMaSS has been programmed in Python scripts with pandas tools (McKinney et al., 2010). As all procedures for generating Polish landscape models for ALMaSS are automated or semi-automated, any landscape in Poland can be 'captured' and used for simulation.

2.1.3. Representative study areas and their landscape characteristics

For this study, we selected nine study areas of 10×10 km^2 with predominant agricultural land use (>60%), and semi-natural habitats covering up to 30% (Fig. 1, Table 1). They represent a gradient from small to large-scale farming, with number of agricultural fields varying from around 1000 fields in study areas LU and RE to more than 8000 fields in WI area (Fig. 1, Table B1 in Appendix B). The study areas also differ in terms of farming systems, with arable farms strongly dominated (~90% of agricultural land) in study areas LU, DO and WA, important (30–40% of agricultural land) share of cattle farms in study areas NG, RE, SK and LI, and pig farms (~20% of agricultural land) in study areas WI and LI, and farming equally divided between arable and cattle production in the study area KR. Total vegetable production represents only a small fraction, with the exception of study areas RE and WI where around 10% of agricultural land is covered by potato farms, and study areas RE and SK where around 5% of agricultural land goes for production of other (besides potato and beet) vegetables. Hobby farms constitute a larger share (around 5% of agricultural land) only in study areas dominated by extremely small fields (WI and SK).

We conducted principal component analysis (PCA) on standardized landscape structure metrics (Table 1) to identify independent components of landscape structure characterizing our study areas. Metrics expressed as percentages were arc sin square root transformed before performing PCA in R environment (R Core Team, 2017). We selected for further analysis components with eigenvalues above 1, i.e., PC1 and PC2, which accounted for 55.4% and 33.0% of the total inertia, respectively (Fig. 2A). The first component (PC1) grouped metrics characterizing farmland structure, while the second component (PC2) captured heterogeneity of off-crop habitats (Fig. 2B).

2.2. Animal model

2.2.1. Overview of the carabid beetle model

The original ALMaSS model for the carabid beetle *B. lampros* was described by Bilde and Topping (2004), and the online ODdox documentation is provided at https://projects.au.dk/fileadmin/dmu.dk/en/animalsplants/almass/ALMaSS/Carabid_B/index.html. Due to the very high number of beetles in the real world, the model uses the super-individual concept, which means that each beetle agent in the model represents 100 real world beetles (Topping et al., 2015). *B. lampros* behavior is simulated on a daily-step. All individuals are categorized as being members of four life-stages: egg, larvae, pupae, and adult female (males are not modeled as they do not limit the population size). Behavior is characterized by annual dispersal and aggregation phases with aggregation linked to non-cultivated habitats and dispersal and breeding largely occurring in open areas. Most important drivers in the model are temperature-controlled developmental rates of eggs, larvae and pupae, together with adult female interactions with the landscape. At high densities the model invokes density dependence via intraspecific predation. In addition, *B. lampros* responds to in-field farming activities, with beetle mortality values for soil cultivation and harvest operations defined based on Thorbek and Bilde (2004).

The response to the pesticide is included in the model by assuming a threshold environmental concentration above which there is a daily probability of mortality (p). This probability is calculated from the following equation:

$$(1-m) = (1-p)^d, \tag{1}$$

where m is the proportion of beetles assumed to die (e.g., 0.9 for 90% mortality over the test period) and d is the number of days over which the test was carried out. If the beetle finds itself in a 1 m^2 grid cell with an environmental concentration above the trigger, then it is assumed to die with the probability p. There is no dose–response, so the maximum death rate is set as m over d days.

Table 1
Landscape structure metrics of the study areas. For each metric, study areas with minimum and maximum values are reported in the parenthesis. Landscape diversity and landscape shape index were computed with FRAGSTATS v4 software (McGarigal et al., 2012).

Metric	Abbreviations [used in Fig. 3]	Explanation	Min	Max
Share of agricultural land	agri_share	% of agriculturally managed land in a study area.	60.3 (SK)	86.4 (DO)
Share of herbaceous semi-natural habitats	herbi_share	% of herbaceous semi-natural habitats in a study area.	5.3 (LU)	19.7 (WA)
Share of woody semi-natural habitats	woody_share	% of woody semi-natural habitats in a study area.	1.2 (DO)	22.2 (RE)
Landscape diversity	LD	Shannon's diversity index (≥ 0, without limit) of landscape element types including six categories: arable land, herbaceous semi-natural habitats, woodland (woody semi-natural habitats), built-up areas, water and others. Shannon's diversity index = 0 when landscape contains only 1 patch (i.e., no diversity), and increases as the number of different patch types increases and/or proportional distribution of area among patch types becomes more equitable.	0.6 (DO)	1.2 (SK)
Landscape shape index	LSI	Normalized ratio of edge (i.e., patch perimeters) to area (class or landscape) in which the total length of edge is compared to a landscape with a standard shape (square) of the same size and without any internal edge. Values greater than one indicate increasing levels of internal edge and corresponding decreasing aggregation of patch types.	57.1 (LU)	144.7 (WI)
Number of agricultural parcels	agri_fields_no	Number of agricultural fields (including permanent crops) in a study area.	973 (RE)	8416 (WI)
Mean agricultural field size	agri_field_size	Mean field size in ha.	0.9 (WI)	7.5 (LU)
Number of farms	farms_no	Number of farms in a study area.	185 (LU)	1146 (WI)
Field boundaries density	bound_dens	Perimeter of field boundaries to total agricultural land [1/m].	0.017 (RE)	0.074 (WI)

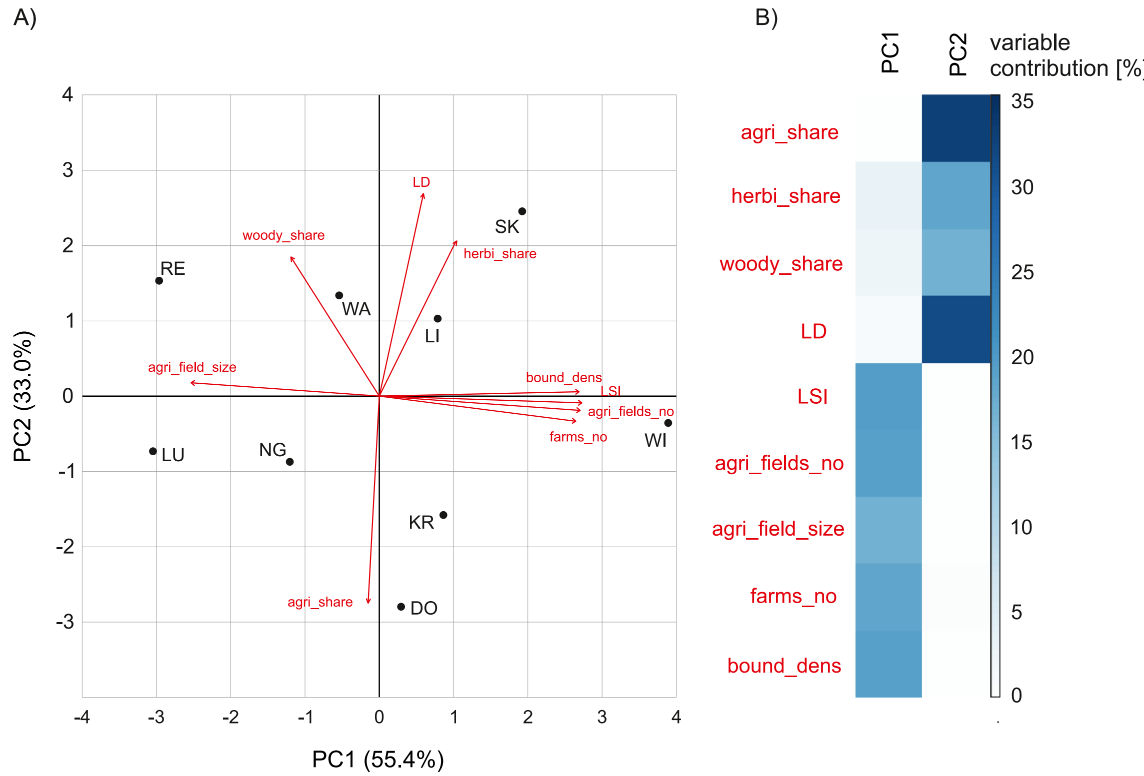

Fig. 2. (A) Biplot of the PCA model conducted for the study areas with the landscape metrics from Table 1. Black dots and lettering discriminate study areas along axes PC1 and PC2. Red arrows and lettering indicate involved metrics. (B) Contributions (in percentage) of the different metrics to the principal components PC1 and PC2. The contribution of a variable to a given principal component is calculated as: (var.cos2 * 100) / (total cos2 of the component), where cos2 represents the quality of representation for variables on the factor map (calculated as the squared coordinates). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

2.2.2. Model calibration

The original ALMaSS model for the ground beetle *B. lampros* was developed and tested using field and phenology data for Denmark, therefore it was necessary to adapt the model to the Polish climatic conditions in respect of critical temperature-dependent processes in the model. Weather daily data (mean air temperature, wind speed, soil temperature, snow cover depth and sum of precipitation) were obtained from the data service of the Polish Institute of Meteorology and Water Management - National Research Institute (https://danepubliczne.imgw.pl/). For model calibration, data from two stations were used: Poznań (station no. 352160330) and Olsztyn (station no. 353200272). The processing of weather data is described in Appendix C.

2.2.2.1. Winter survival. In the model, winter survival of *B. lampros* is a function of sum of negative degrees accumulated during overwintering (Petersen et al., 1996) (Fig. D1 in Appendix D):

$$x = \Sigma\ negative^{\circ}C \tag{2}$$

if $x > -40$ *then survival chance [%/day]* $= 0.94925 + x \times 0.00426$

if $-40 \geq x > -80$ *then survival chance [%/day]* $= 1.16913 + x \times 0.01149$

if $-80 \geq x > -107$ *then survival chance [%/day]* $= 0.6665 + x \times 0.00528$

else mortality chance [%/day] $= 0.1$

When, according to the original model, air temperatures were used to calculate winter survival of beetles, population was severely reduced in Polish climatic conditions. For example, mean winter survival for years 2010–2019 calculated using data from the Poznań station (central Poland) was only 17.4% compared to 72.9% when data from the Sandvig station in Denmark were used (Fig. D2 in Appendix D). However, during overwintering when beetles hide under the vegetation and snow or even in the surface layer of soil, they do not experience the same low temperatures as measured two meters above the ground by standard meteorological stations. In the winter, soil temperatures tend to be higher and fluctuate less than air temperatures. Because winters in Denmark are generally much warmer than in Poland, using air temperatures to calculate winter survival in the model for Denmark did not affect the winter survival of beetles as considerably as in Poland. Therefore, we decided to use soil temperature at depth of 5 cm instead of air temperature to calculate winter survival of beetles in the Polish climatic conditions. This solution resulted in raising mean winter survival for years 2010–2019 from 17.4% to 46.2% (Fig. D2 in Appendix D).

2.2.2.2. Duration of developmental stages. In the original model, temperature-dependent development is calculated for each developmental stage: egg, larva, and pupa based on data from (Jensen, 1990). Development *f(dev)* is calculated as:

$$f(dev)\,(T) = T - T0/L \tag{3}$$

where *T* is air temperature, *T0* is lower threshold for development, and *L* is duration of stage in day degrees. As *B. lampros* development from egg to pupa occurs in the surface layer of soil, we used soil temperature at depth of 5 cm instead of air temperature also in calculations of temperature-dependent development according to Eq. [3].

2.2.2.3. Spring dispersal and reproduction triggers. We fitted the original beetle model to the phenology data for the *B. lampros* from the study areas from Poland (located near Poznań and Olsztyn) provided by G. Sowa and A. Kosewska (personal communication). This resulted in setting the onset of spring dispersal to 8 degree days (accumulated above the threshold of 6 °C) reached after March 1st (with no dispersal allowed before March 1st), and egg laying temperature threshold to 6 °C (see details in Appendix C).

2.3. Simulation scenarios

We implemented two types of mitigation measures in ALMaSS, (1) in-crop measures related to substitution of harmful insecticides by less harmful insecticides, and (2) off-crop measures related to increase of grassy field margins in a landscape. The first type of measure can be considered equivalent to a reduction in exposure or application rate, or toxicological impact of insecticides in the field, while second one helps to reduce the exposure to off-crop areas, such as field ditches, and provides additional overwintering, reproduction or foraging habitats for modeled species.

2.3.1. In-crop mitigation measures: reducing lethality caused by insecticides

We assumed that normal fungicide and herbicide applications have no impact on carabid beetles. For the insecticides, we considered five insecticide-driven beetle field lethality rates (LR) decreasing from 90% to 10% by 20%, as measured for a foliar insecticide-spray application over seven days. This results in daily beetle probability of mortality *p* decreasing from 0.28 to 0.01. Decreasing probability of field mortality simulates less toxic insecticides or lower doses used, but could also be considered to result from decreased sensitivity of the beetles to the applied insecticide.

In simulations, two significantly different values for environmental degradation time (DT_{50}) at 20 °C, 3 and 25 days, were chosen based on field measurements of DT_{50} (according to the Pesticide Properties DataBase of the University of Hertfordshire; https://sitem.herts.ac.uk/aeru/ppdb/) for insecticides most commonly used in Poland. DT_{50} of 25 days is characteristic for e.g. Lambda-cyhalothrin and cypermethrin, while DT_{50} of 3 days for e.g. acetamiprid. The temperature dependence of the DT_{50} value was defined according to the following equation (EFSA, 2007):

$$DT_{50}(t) = DT_{50}(20^{\circ}) \times exp\left[0.094779 \times (20^{\circ} - t)\right] \quad (4)$$

where t is a given mean daily temperature.

To ensure that beetles could be exposed above the trigger threshold for at least the period defined by DT_{50} (3 or 25 days), a treatment rate of 5.05 of the trigger concentration at each LR was used for insecticides with $DT_{50} = 3$ days, and a treatment rate of 1.22 of the trigger concentration at each LR was used for insecticides with $DT_{50} = 25$ days. These rates (5.05 and 1.22) were calculated as follows:

$$treatment\ rate = 1/\left[0.5^{1/DT_{50}\ [days]} \times duration\ of\ effect\ [days]\right] \quad (5)$$

In all scenarios, we considered spray drift up to 12 m from the edge of any sprayed field, following the equation by Rautmann et al. (2001) with the reduction of 50%, which can be considered as a minimum requirement for spraying equipment in field crops using modern spraying equipment (JKI, 2020).

2.3.2. Off-crop mitigation measures: introducing additional field margins

As field margins (understood as in-field grassy linear structures not subjected to the same agricultural practice as the crop itself such as ploughing or harvesting or pesticide application) in the Polish agricultural landscapes are usually very narrow (20–50 cm) they are rarely mapped in the BDOT database. After visual examination of the study landscapes, we artificially manipulated our landscape maps to include 1-m field margins to 20% of fields in each of the study area. Furthermore, for a subset of scenarios we increased the proportion of fields with field margins from 20% to 40% and 60%, and increased their width from 1 m to 2 m and 4 m. Field margins were only added to the fields that were bigger than 1 ha and wider than 20 m, since according to field observations, field margins of 1 m width are rarely found in these smaller fields. In addition, field margins were not added in places where the field already bordered grassy margin or other herbaceous habitat, such as managed or unmanaged grassland.

2.3.3. Simulation set-up

Each combination of the nine study areas, five insecticide-driven lethality rates, two insecticides environmental decay rates, three field margins levels of presence in the landscape, and three field margins widths was tested for its effect on population dynamic of *B. lampros* (altogether 810 combinations). The simulation for each combination was replicated 10 times, as this number of replicates was shown to be sufficient to account for between-replicates variability, which in case of the *Bembidion* model is generally very low (Topping et al., 2015). Initial conditions for each replicate differed in the distribution of beetles across the study area and the initial allocation of crops in the fields, but not in the starting number of super-individuals which was set to 200,000 per 100 km^2. All combinations' runs were for 40 years, with the data analyzed based on the mean result of the last 30 years of simulation. Data from the first 10 years of each simulation was ignored, as this was the period needed for the population to stabilize in the model. Weather conditions were selected to represent the 2010–2019 period from the Poznań station, located near the study areas (Fig. 1), and the weather cycle was repeated after each 10 years. For details regarding weather data processing see Appendix B.

2.3.4. Simulation outputs and data analysis

From each simulation three endpoints were analyzed: mean overall beetle population density (total number of adult female beetles divided by the landscape area, i.e. 100 km^2), occupancy (proportion of grid cells in the landscape with at least 100 adult female beetles) and mean abundance (mean number of adult females in the occupied cells). The latter two endpoints are presented as Abundance-Occupancy Relationship (AOR) plots (Høye et al., 2012). Although the spatial resolution of landscape model in ALMaSS is 1 m^2, for occupancy and abundance calculation grid cells of 50 m^2 were used. Overall population density, and beetle abundance and occupancy were measured at day 59 of each year (March 1st) of each simulation, and then means over last 30 years of the simulations were calculated. These endpoints were then averaged across 10 replicate runs for each scenario. To determine the effect size of each mitigation measure on *B. lampros* population, the impact of each scenario relative to the baseline was used and compared over time. We set baseline conditions according to the 'worst-case' scenario with LR of 90%, DT_{50} of 25 days (longer persistence of insecticides in the environment), and field margins of 1-m width added to only 20% of fields.

The influence of components of landscape structure (PC1 and PC2) on mean overall beetle density in the baseline ('worst-case') scenario and on changes of mean overall beetle density in response to applied mitigation measures was tested with general linear models (GLMs). Calculations were performed in R environment (R Core Team, 2017). Separate GLMs were constructed for toxicity-related and field margins-related scenarios, as well as for scenarios with DT_{50} of 3 and 25 days. For each GLM model, we reported Cox and Snell pseudo *R-squared* value, as well as *p*-value calculated with the likelihood ratio test.

3. Results

The impacts of in-crop and off-crop mitigation measures on the population of *B. lampros* in different landscape settings scenarios was simulated with ALMaSS. For each scenario, coefficients of variation (cv) for the simulation endpoints were computed from 10 replicate runs and were in all cases not greater than 2.64% for the mean overall population density, 1.63% for the mean abundance and 3.61% for the mean occupancy. Simulation replicates were therefore very similar and no more than 10 replicates were needed.

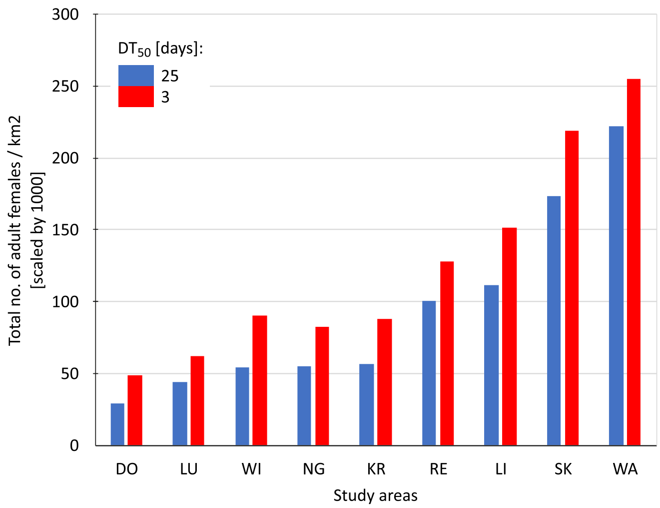

Fig. 3. The mean overall population density of *B. lampros* in nine study areas according to the baseline ('worst-case') scenario with LR of 90%, DT_{50} of 25 days (longer persistence of insecticides in the environment), and field margins of 1-m width added to only 20% of fields (blue). For comparison the mean overall beetle density with decreased DT_{50} of 3 days is shown (red). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

3.1. Beetle populations in baseline 'worst-case' scenario

The beetle populations in the baseline 'worst-case' scenario varied across study areas (Fig. 3), strongly depending on landscape heterogeneity (Table E1 in Appendix E). The mean overall beetle density increased significantly with PC2 component ($p = 0.01$), i.e., the more semi-natural habitats and the higher the landscape heterogeneity in the study area, the greater beetle population was found, but was not influenced by PC1 component ($p = 0.72$). The model was significant at $p < 0.01$ and explained 66% of total variance. Note that when the DT_{50} was decreased to 3 days, the mean overall beetle density was significantly influenced by both PC1 ($p < 0.01$) and PC2 ($p < 0.01$) landscape structure components, i.e., the beetle populations were better supported in study areas with high heterogeneity of both landscape and farmland (model was significant at $p < 0.01$ and explained 88% of total variance).

The between-years variation in beetle densities was strongly related to changes in weather, with repeating 10-year cycles being clearly visible (Fig. 4). In the 'worst-case' scenario beetle populations in all study areas tended towards extinction, but at different rates. Mean overall beetle density measured between like weather years (10 years apart) decreased with simulation time, and showed considerable variation among in-between like weather years for all study areas (Fig. E1 in Appendix E).

3.2. Applying of mitigation measures

The impacts of applying each mitigation measure separately are described below (reducing lethality caused by insecticides in Section 3.2.1, and introducing additional field margins in Section 3.2.2), and then the combined effects are shown (Section 3.2.3). All the results described in Sections 3.2.1–3.2.3 refer to scenarios in which DT_{50} was set to 25 days (results for scenarios with DT_{50} of 3 days can be found in Appendix E). Comparison with results where DT_{50} of 3 days was used, i.e., the impact of decreasing the persistence of insecticides in the environment, is presented in Section 3.2.4.

3.2.1. Reducing lethality caused by insecticides

The mean overall beetle density, abundance and occupancy calculated for scenarios with different insecticide-driven lethality rates differed considerably among the study areas. Decreasing the insecticide-driven lethality rate from LR90 (baseline 'worst-case' scenario) to LR10 resulted in an exponential increase in mean overall population density in all study areas (Fig. E2 in Appendix E). The relative change in mean overall population density was weakly influenced by PC1 (farmland structure; $p = 0.03$) but decreased significantly with PC2 component ($p < 0.01$), meaning that the impact of reduction of insecticide-driven lethality on beetle population was higher in study areas with small proportion of semi-natural habitats and low landscape heterogeneity (Table E3 in Appendix E; model was significant at $p < 0.01$ and explained 96% of total variance). Therefore, the strongest impacts were found in the group of study areas with baseline beetle populations at the level < 75,000 adult females per km^2 (NG, KR, LU, DO and WI). In all these study areas decreasing insecticide-driven lethality rate by only 20% (from LR90 to LR70) resulted in more than 10% increase in mean overall population density, and further reduction of insecticide-driven lethality rate to LR50 caused an over 30% increase

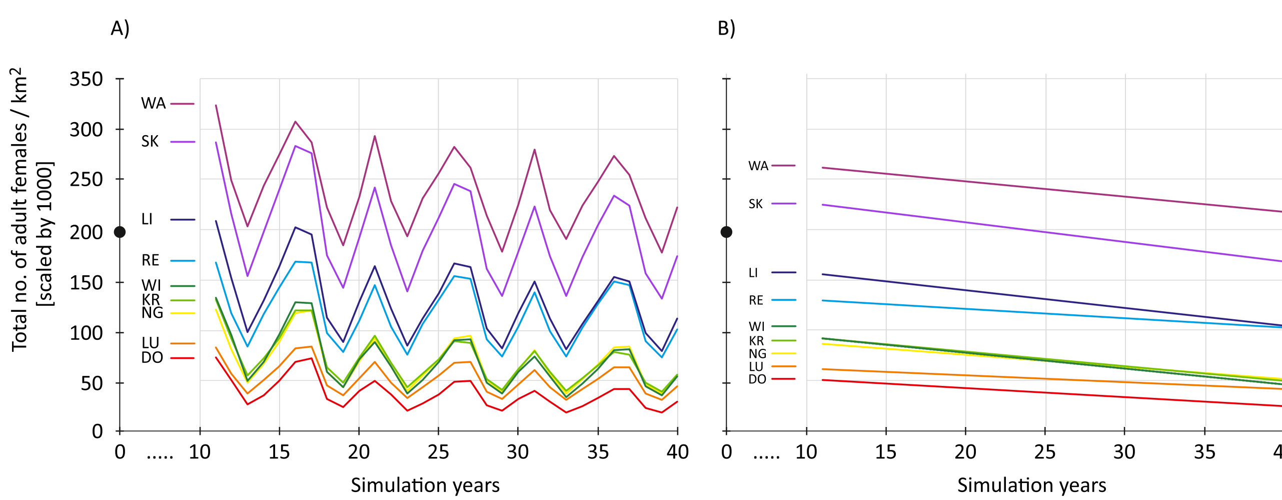

Fig. 4. Between-years variation (A) and trend lines (B) for beetle densities in the 'worst-case' scenario (averaged across 10 replicate runs) as measured at March 1st of each simulation year for the nine analyzed study areas. 'Worst-case' scenario was defined as scenario with LR of 90%, DT_{50} of 25 days (longer persistence of insecticides in the environment), and field margins of 1-m width added to only 20% of fields. Only results for years 11–40 of simulation are shown (ignoring first 10 years during which populations were stabilizing). Black dot at Y axis marks the starting number of adult females in each simulation (200,000 per km^2). Note that in study areas WI, KR, NG, LU and DO beetle densities even in the best years rarely exceeded 100,000 adult females per km^2. The colors assigned to study areas are used consequently in all following figures. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 2
Relative changes (%) in comparison to the baseline (scenario with LR of 90%, DT_{50} of 25 days, and field margins of 1-m width added to only 20% of fields) in beetle densities for scenarios with different insecticide-driven lethality rates. The order of the study areas follows the increasing overall beetle density in the baseline scenario (Fig. 3).

Study area	*LR70*	*LR50*	*LR30*	*LR10*
DO	20.2	56.5	116.8	528.3
LU	11.7	31.8	69.5	274.5
WI	14.8	38.6	77.6	326.1
NG	12.0	34.2	70.6	261.3
KR	13.6	37.5	74.5	292.0
RE	6.3	17.3	35.4	127.5
LI	7.6	20.9	42.2	156.0
SK	5.3	14.1	28.6	104.5
WA	2.8	8.8	17.5	67.7

Reaching a goal of increasing beetle density by:
< 10
[10, 20)
[20, 50)
[50, 100)
≥ 100
% in relation to the baseline scenario

of the beetle populations (Table 2). On the other hand, the impact of decreasing insecticide-driven lethality on the increase in population density in the study areas with high proportion of herbaceous habitats and high landscape diversity, with high initial beetle density (baseline density > 100,000 adult females per km^2 in RE, LI, SK and WA), although still visible, was much smaller and limited by the density-dependence mechanism, especially in the scenario with the lowest LR10 (Table 2).

In all scenarios reducing insecticide-driven lethality rate, the higher the overall population density the higher the mean beetle occupancy in all study areas (Fig. 5). In contrast, reducing the insecticide-driven lethality rate up to LR50 (for study areas DO, KR, NG, WI and LI) or even LR30 (for study areas RE, LU, WA, SK) had a negative impact on mean beetle abundance (Fig. 5), being a consequence of individuals expanding into new areas but with still low carrying capacity. Substantially higher mean beetle abundance than in the baseline scenario was reached in scenario LR10 in all study areas besides RE and WA. Reducing insecticide-driven lethality rate to LR10 in study areas with the lowest baseline mean beetle abundances resulted in an increase in abundance exceeding 45% (in WI and KR) or even reaching 70% in the DO study area (Fig. 5).

3.2.2. Introducing additional field margins

In a subset of scenarios field margins of 1 m, 2 m or 4 m were added to different proportions of agricultural fields (20%, 40% or 60%). Because study areas differed in terms of farmland heterogeneity (number and size of fields), the same field margin scenario could result in generation of different proportion of field margins in a landscape depending on a study area (Table E4 in Appendix E). For example, generation of field margins of 2-m width to 40% of fields (scenario 'FM40w2m') resulted in 6.6% of WI study area being covered by field margins but only 1.6% in case of RE study area. In addition, in study areas WI, SK, LI and WA it was not possible to apply scenarios with field margins added to 60% of agricultural fields (scenarios 'FM60w1m', 'FM60w2m' and 'FM60w4m') because these areas did not have 60% of fields bigger than 1 ha and wider than 20 m (according to constraints for field margins generation described in Section 2.3.2).

If insecticide-driven lethality rate was kept constant, increasing the percentage of field margins of 1-m width from 20% (baseline 'FM20w1m') to 40% (scenario 'FM40w1m') had negligible effect (increase up to 3%) on the mean overall beetle density (Table 3). The mean beetle occupancy was affected more substantially, with the highest increase noted in study areas KR and DO (~10%). Further increase in field margins abundance to 60% of fields (scenario 'FM60w1m') resulted in an increases in the mean overall beetle density by up to 5% (Table 3) and allowed an increase in the baseline mean beetle occupancy by up to 15%. Relatively low increases in mean overall beetle density combined with more substantial increases in mean beetle occupancy resulted in negative changes in mean beetle abundance, up to 6% when percentage of fields with field margins was increased from 20% to 40%, and up to 8% with further increase of abundance of field margins up to 60%.

Increasing the width of field margins from 1 m (baseline 'FM20w1m') to 2 m (scenario 'FM20w2m') had negligible effect on mean overall beetle density (Table 3) and on AORs. Further increase of field margins width to 4 m (scenario 'FM20w4m') resulted in increases of mean overall beetle density up to 7%, increases in mean beetle occupancy up to 10%, and decreases of mean beetle abundance up to 3% in relation to the baseline (Table 3).

We also investigated the cumulative impact on beetle populations when both off-crop measures were applied together (i.e., the abundance and width of field margins were increased). The biggest impacts on beetle populations had scenarios in which the percentage of fields with margins was increased to 40% or, if possible, 60%, and field margins width to 4 m (scenarios 'FM40w4m' and 'FM60w4m'). This resulted in at least 9% increase in the mean overall beetle density (Table 3), and at least 16% increase in the mean beetle occupancy and decrease in mean beetle abundance bigger than 6% (Fig. 6) in the study areas with low baseline beetle populations (NG, KR, LU, DO and WI). In general, introducing field margins of 4-m width to 40% or 60% of fields gave comparable impacts on mean overall beetle density and occupancy to decreasing the insecticide-driven lethality rate to LR70. However, none of the off-crop measures scenarios tested resulted in an impact comparable to those resulting from decreasing the insecticide-driven lethality rate to LR50.

The increase in the overall mean beetle density in response to increasing abundance of field margins (being a result of applying different field margins scenarios) differed among study areas (Fig. 7) and was driven by landscape and farmland heterogeneity. Slopes of the regression lines significantly decreased with both PC1 ($p = 0.01$) and PC2 ($p < 0.01$), meaning that the impact of introducing field margins on beetle population decreased with the share of semi-natural habitats, landscape diversity and farmland heterogeneity (was lowest in study areas with dominance of small fields, high density of field boundaries and low aggregation of patches; Table E5 in Appendix E). The model was significant at $p < 0.01$ and explained 83% of total variance.

3.2.3. Combining mitigation measures

Combining mitigation measures generally magnified the effects of single measure application. However, those impacts were not simply additive and strongly differed among the study areas (Table 4, Fig. E5 in Appendix E), indicating the importance of landscape structure. In general, the biggest positive relative impact of reducing the lethality caused by insecticides while simultaneously increasing field margins was noted for the lethality rate change from LR90 to LR70 (Table 4). In that case, even an increase of field margins abundance, with no change in width, from 20% to 40% increased positive impact on mean overall beetle density by at least 10% (except for landscapes SK and WA), and mean beetle occupancy by at least 20%. If field margins width was increased to 2 m, it increased the positive impacts by at least 20% on mean overall beetle density, and at least 40% on mean beetle occupancy (Table 4). More importantly, the positive impacts on beetle populations resulting from increasing field margins abundance to 40% or 60% and their width to 4 m together with reducing the insecticide-driven lethality rate to LR70, were comparable to reduction of insecticide-driven lethality rate to LR50 (Table 4).

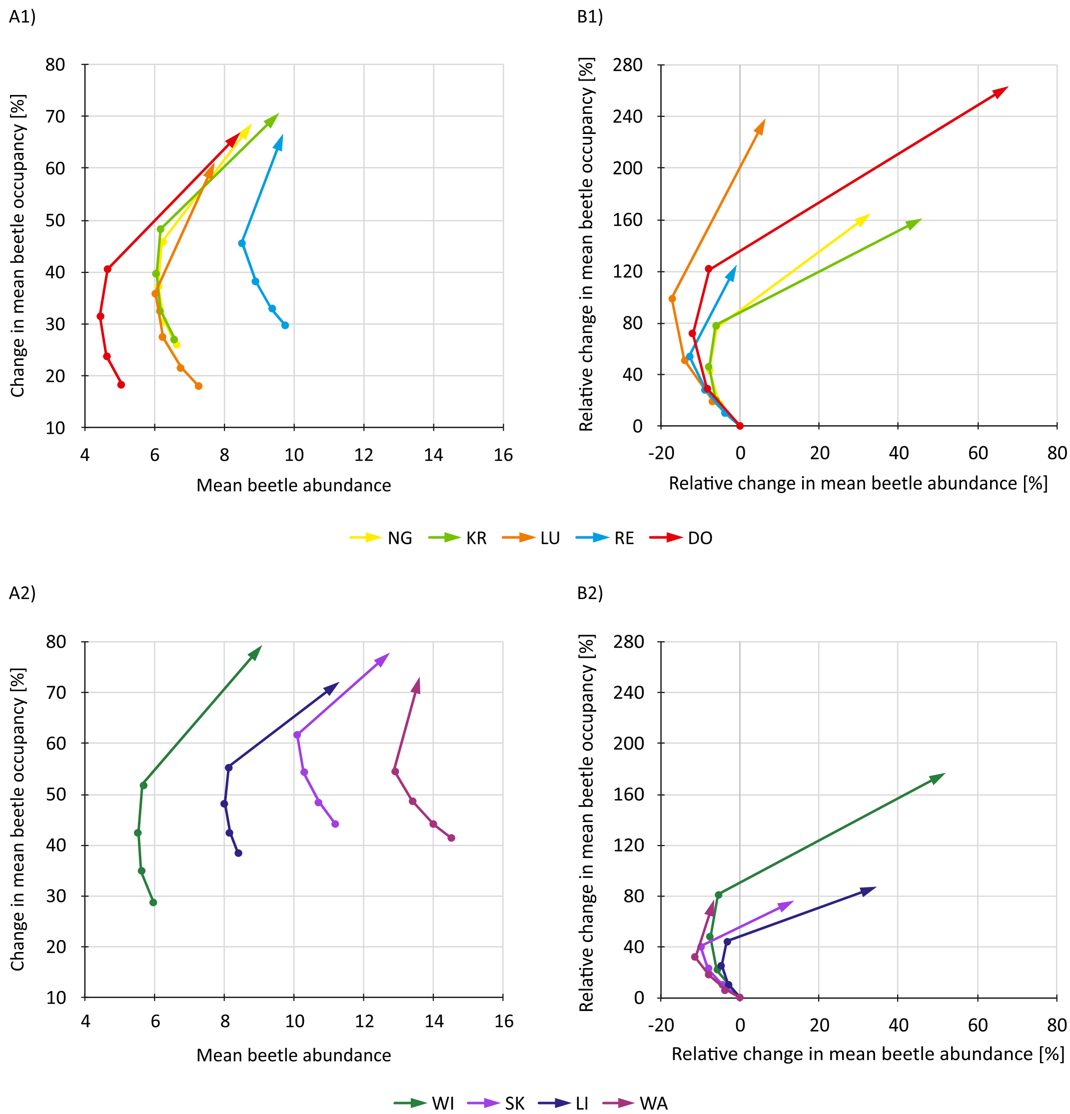

Fig. 5. Changes in mean beetle occupancy and abundance of *B. lampros* populations (plot of Abundance to Occupancy Relationship, AOR) in scenarios decreasing the insecticide-driven lethality rate and with DT_{50} of 25 days: (A) absolute changes, (B) relative changes in comparison to the baseline scenario (with LR of 90%, DT_{50} of 25 days, and field margins of 1-m width added to only 20% of fields) for study areas NG, KR, LU, RE, DO (1), and study areas WI, SK, LI and WA (2). Dots mark consecutively, starting from the bottom, the baseline scenario (0,0), and changes in AOR resulted from decreasing the insecticide-driven lethality rate to LR70, LR50, and LR30.

Table 3
Relative changes (%) in the overall mean beetle density in different field margins scenarios in relation to the baseline (scenario with LR of 90%, DT_{50} of 25 days, and field margins of 1-m width added to only 20% of fields). For easy comparison, cell shading follows the same scale as in Table 2; grey cells (N/A, data not available) indicate study areas for which it was not possible to find sufficient number of fields to increase field margin abundance to 60%. The order of the study areas follows the increasing overall beetle density in the baseline scenario (Fig. 3).

Study area	*FM40w1m*	*FM60w1m*	*FM20w2m*	*FM20w4m*	*FM40w2m*	*FM60w2m*	*FM40w4m*	*FM60w4m*
DO	2.9	5.1	1.2	6.9	6.6	12.2	19.0	28.3
LU	3.1	3.4	1.9	4.7	3.9	7.6	9.5	14.2
WI	0.6		1.5	4.8	3.8		11.5	
NG	2.3	3.5	-0.2	3.1	4.1	7.5	9.7	15.5
KR	2.8	4.1	1.7	5.4	6.2	9.0	10.8	15.9
RE	1.6	2.9	1.1	2.8	2.7	5.8	5.4	8.8
LI	0.7		0.4	2.3	2.6		6.2	
SK	0.0		0.9	1.8	1.3		3.6	
WA	0.0		0.1	0.4	0.4		2.1	

Reaching a goal of increasing beetle density by:
< 10
[10, 20)
≥ 20
N/A
% in relation to the baseline scenario

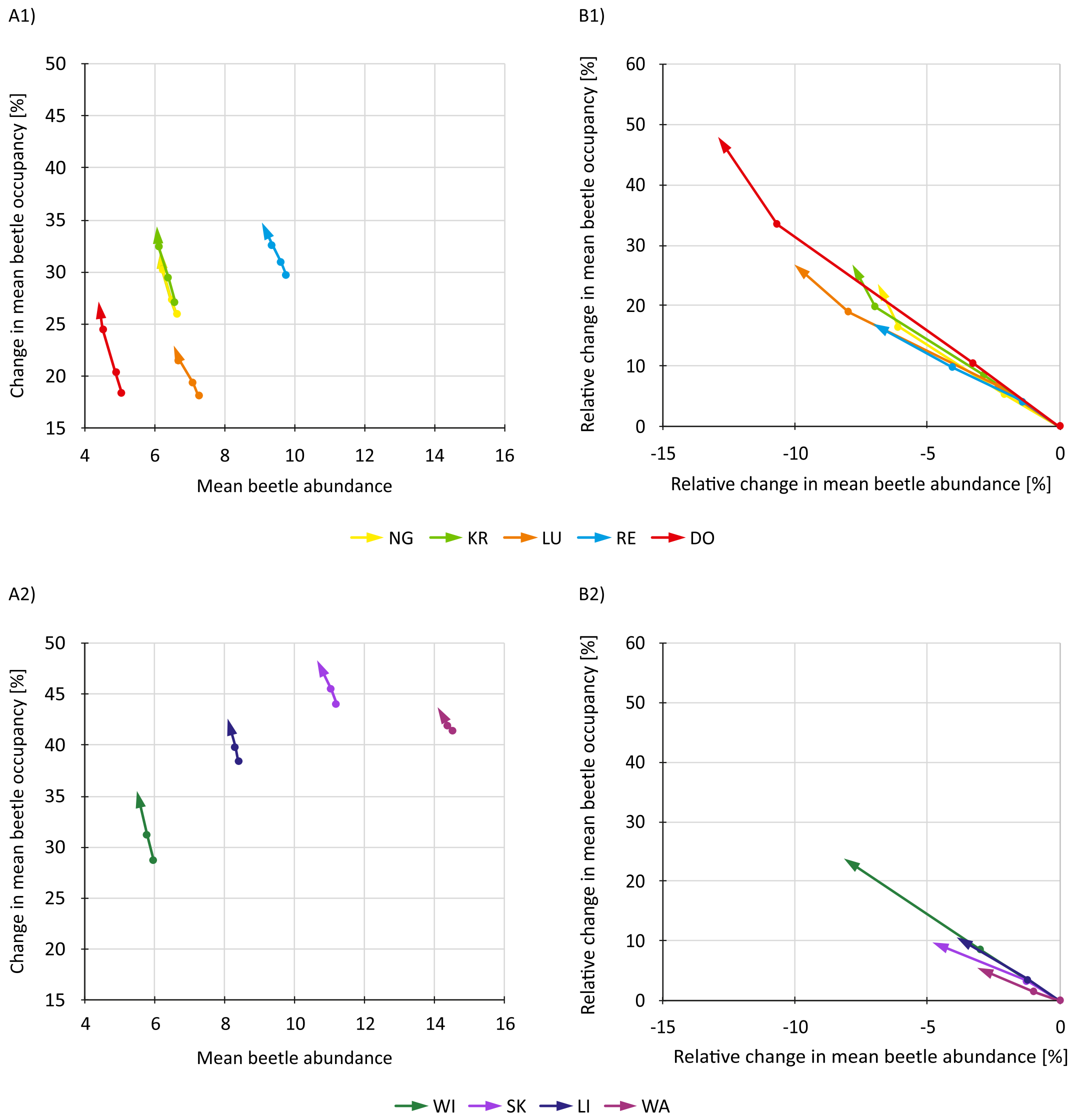

Fig. 6. Changes in mean beetle occupancy and abundance of *B. lampros* populations (plot of Abundance to Occupancy Relationship) in response to selected field margins-scenarios: (A) absolute changes, (B) relative changes in comparison to the baseline scenario (with LR of 90%, DT_{50} of 25 days, and field margins of 1-m width added to only 20% of fields). In study areas NG, KR, LU, DO and RE (1) dots mark consecutively, starting from the bottom, the baseline scenario (0, 0), and changes in AOR resulted from the increase in percentage of fields with 4-m width from 20% ('FM20w4m') to 40% ('FM40w4m'), and 60% ('FM60w4m'). In study areas WI, SK, LI and WA (2) dots mark consecutively, starting from the bottom, the baseline scenario (0, 0), and changes in AOR resulted from the increase in percentage of fields with 4-m width from 20% ('FM20w4m') to 40% ('FM40w4m'). For those study areas it was not possible to apply scenario 'FM60w4m' as it was not possible to find sufficient number of fields to increase field margin abundance to 60%.

3.2.4. Influence of DT_{50}

Reducing DT_{50} from 25 days to 3 days in the 'worst-case' scenario resulted in positive impacts on mean overall beetle density (increase by 6% to 31%, depending on the study area) and occupancy (increase by 12% to 40%, depending on the study area), but had negative impacts on mean beetle abundance (decrease by 3% to 8%, depending on the study area) in all tested scenarios. When the insecticide-driven lethality rate was decreased up to LR30, the positive impacts on mean overall beetle density were more than 50% higher in scenarios with DT_{50} of 3 days than DT_{50} of 25 days, but only up to 20% higher when the lethality rate was further reduced to LR10. Reducing DT_{50} from 25 days to 3 days magnified also increases in mean beetle occupancy when the insecticide-driven lethality rate was decreased to LR70 and LR50. When reducing the lethality rate to LR30, the effect of DT_{50} was not visible, and in scenarios with reduction of the lethality rate to LR10 higher positive impact on mean beetle occupancy was found when DT_{50} was set to 25 days (Fig. 8). In case of abundance, when reducing the insecticide-drive lethality rate the earlier shift from decreasing to increasing of abundance could be noticed in scenarios where DT_{50} was 3 days (Fig. 8).

Decreasing the DT_{50} from 25 days to 3 days magnified the positive impacts of field margins on the mean overall beetle density and decreased the negative impacts on mean beetle abundance. However, the influence of decreasing DT_{50} from 25 days to 3 days on mean beetle occupancy was negligible.

4. Discussion

Our findings fill an important gap in the evidence base on effects of different mitigation measures on population dynamic of non-target arthropods in landscapes of varying heterogeneity. We showed that mitigation of agricultural intensification by both in-crop and off-crop measures can positively affect the population dynamics of the widespread carabid beetle *B. lampros*, but the strength of the effect differs depending on landscape heterogeneity (mainly the abundance of semi-natural habitats and landscape diversity).

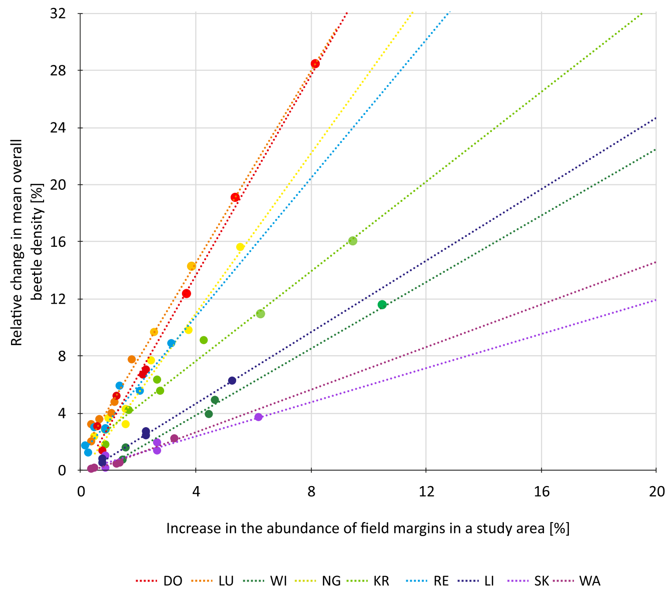

Fig. 7. Relative changes, in comparison to the baseline (scenario with LR of 90%, DT_{50} of 25 days, and field margins of 1-m width added to only 20% of fields) in the mean overall beetle density in response to increasing abundance of field margins in different landscapes.

4.1. Comparing effects of applied mitigation measures

Reducing the lethality caused by insecticides had clearly bigger positive impacts on beetle populations than increasing the abundance of field margins in a landscape. With reduced insecticide-driven lethality we observed substantial increases in mean overall beetle density and (even larger) in mean beetle occupancy, which together resulted in decreasing mean beetle abundance in occupied areas in scenarios with insecticide-driven lethality rate of LR70 and LR50. This is the effect of the insect's range expanding (where the probability of mortality decreased due to decreased lethality caused by insecticides) but with lower carrying capacity. When the insecticide-driven lethality rate was further reduced to LR30 or even LR10, the mean beetle abundance started to increase, exceeding at some point its value from the baseline scenario. The level of reduction in insecticide-driven lethality needed for this shift to occur depended on the degradation time of insecticides, i.e., the less persistent the insecticides are in the environment, the smaller reduction of lethality was needed for mean beetle abundance to exceed the baseline values. The degradation time of insecticides also influenced the magnitude of impacts observed in reduced-lethality scenarios, with more than 50% higher mean overall beetle density found in scenarios where DT_{50} was set to 3 days (besides scenario LR10).

The impacts of increasing grassy field margins depended on their width and overall abundance in the landscape. As field margins 1–2 m wide were impacted by the pesticide drift, they were not able, as a stand-alone measure, to overcome negative effects of high lethality caused by insecticides, especially if only applied to 20%–40% of agricultural fields in studied landscapes. Only field margins 4 m wide, applied to at least 40% of fields in a landscape, resulted in an increase in beetle population density comparable to scenario when the insecticide-driven lethality rate was reduced to LR70. This is somewhat worrying as in many landscapes with intensive agricultural use field margins below 3 m width are most common (study from Germany, Hahn et al., 2014). On the other hand, new technologies with shielded nozzles can reduce the pesticide drift by

Table 4
Proportional changes (%) in relation to the baseline scenario (LR of 90%, DT_{50} of 25 days, and field margins of 1-m width added to only 20% of fields) in the overall mean beetle density when combined mitigation measures were applied. Cell shading follows the same scale as in Table 2; grey cells (N/A, data not available) indicate study areas for which it was not possible to find sufficient number of fields to increase field margin abundance to 60%. The order of the study areas follows the increasing overall beetle density in the baseline scenario (Fig. 3).

Study	*FM40w1m*				*FM60w1m*				*FM20w2m*				*FM40w2m*			
area	*LR70*	*LR50*	*LR30*	*LR10*	*LR70*	*LR50*	*LR30*	*LR10*	*LR70*	*LR50*	*LR30*	*LR10*	*LR70*	*LR50*	*LR30*	*LR10*
DO	25.6	61.8	116.5	523.7	28.4	64.8	118.2	520.9	21.5	58.6	119.7	535.3	31.8	68.2	123.5	541.3
LU	13.8	36.0	73.5	292.9	15.8	38.9	78.8	312.2	11.4	33.1	68.2	281.3	17.5	40.3	79.3	311.0
WI	16.3	38.2	72.7	314.5					16.4	40.7	78.0	335.5	19.5	39.9	76.0	329.7
NG	15.5	37.4	73.6	268.8	17.8	39.1	75.5	272.0	13.5	36.1	73.3	265.0	18.3	41.3	77.5	278.9
KR	17.0	38.7	72.0	290.0	18.0	39.4	71.8	287.3	16.1	39.1	76.0	299.3	20.0	41.6	76.5	299.9
RE	8.5	19.8	41.4	141.3	10.1	23.4	44.0	150.9	6.8	17.7	37.4	130.5	9.9	23.7	43.4	146.9
LI	8.4	20.6	40.8	153.1					8.7	22.4	44.0	159.2	10.5	22.1	42.7	159.8
SK	5.5	13.9	26.6	101.0					6.0	14.8	29.5	107.0	6.5	14.9	28.6	104.9
WA	2.9	9.1	17.8	67.9					3.1	9.0	18.4	69.0	3.5	9.5	18.4	70.5

Study	*FM60w2m*				*FM20w4m*				*FM40w4m*				*FM60w4m*			
area	*LR70*	*LR50*	*LR30*	*LR10*	*LR70*	*LR50*	*LR30*	*LR10*	*LR70*	*LR50*	*LR30*	*LR10*	*LR70*	*LR50*	*LR30*	*LR10*
DO	36.8	71.2	124.2	548.9	27.4	64.7	125.2	564.8	42.5	74.5	133.0	584.5	51.4	82.3	140.9	607.7
LU	20.3	44.4	84.2	326.5	16.6	38.1	78.2	308.4	22.9	48.0	90.9	336.3	30.9	53.8	100.0	359.7
WI					20.0	43.3	83.9	353.6	24.8	44.5	86.1	366.7				
NG	20.9	44.4	80.4	285.9	16.6	39.0	77.6	274.2	24.5	46.1	86.0	298.3	29.8	53.4	92.2	315.7
KR	22.0	42.8	75.1	301.9	19.4	42.5	81.6	312.6	24.5	45.3	82.8	326.2	27.1	49.4	86.3	337.2
RE	12.3	26.3	47.6	156.6	9.6	21.4	39.8	134.3	13.3	26.9	47.8	155.3	17.7	30.6	54.3	172.8
LI					9.8	23.7	45.5	165.5	12.8	25.4	47.2	170.5				
SK					7.2	16.2	31.7	111.8	8.4	16.9	31.6	113.9				
WA					3.8	9.9	19.2	71.9	5.7	11.3	21.3	76.5				

Relative change in beetle density by:
< 10
[10, 20)
[20, 50)
[50, 100)
≥ 100
N/A
% in relation to the baseline scenario

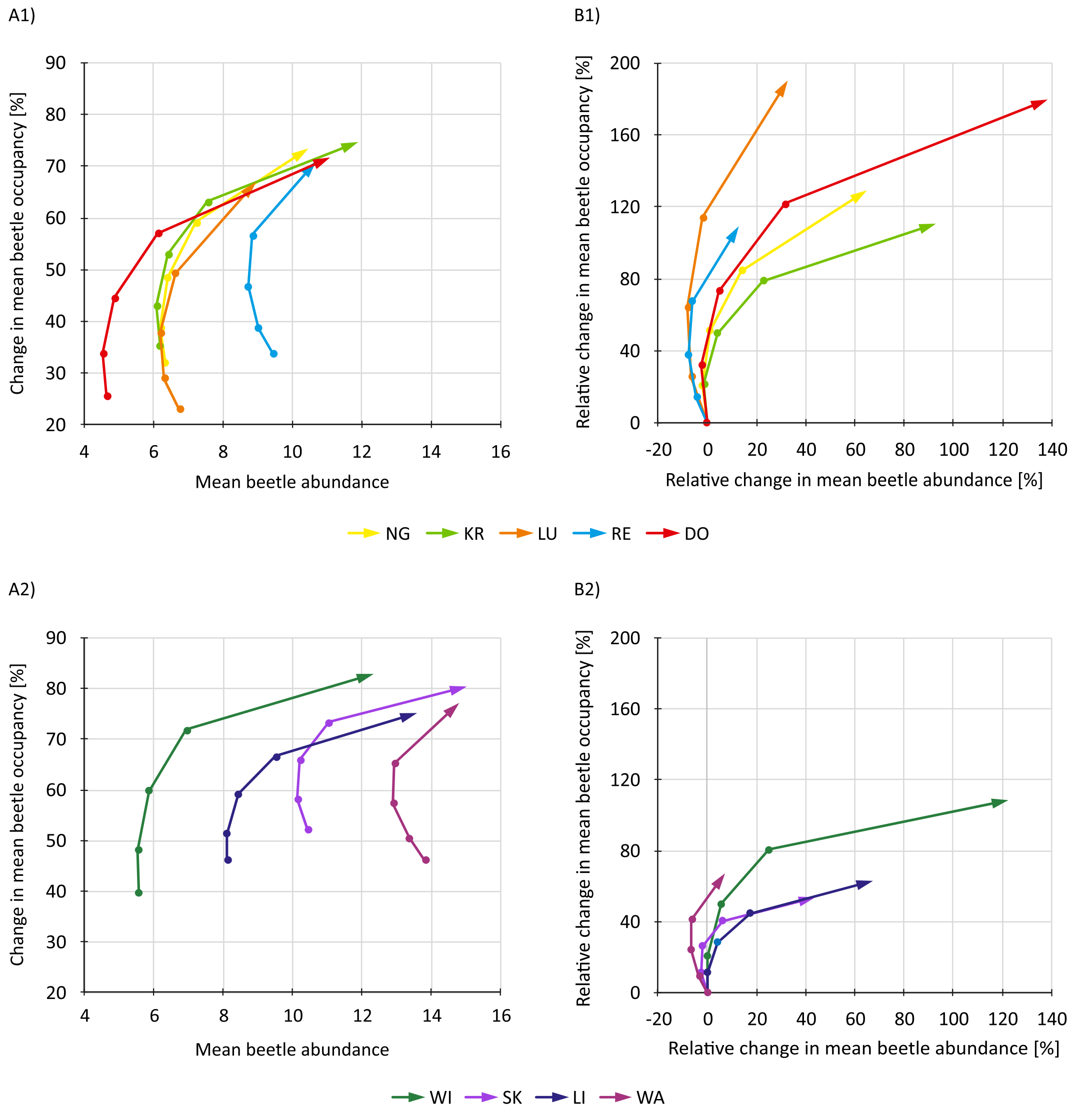

Fig. 8. Changes in mean beetle occupancy and abundance of *B. lampros* populations (plot of Abundance to Occupancy Relationship, AOR) in scenarios decreasing the insecticide-driven lethality rate and with DT_{50} set to 3 days: (A) absolute changes, (B) relative changes in comparison to the scenario with LR of 90%. Dots mark consecutively, starting from the bottom, the baseline scenario (0,0), and changes in AOR resulted from decreasing the insecticide-driven lethality rate to LR70, LR50, and LR30.

more than 90% (JKI, 2020), while we assumed 50% drift reduction. Hence, there is a good chance that with broader introduction of such modern equipment, the field margins below 3 m will serve the purpose. Generally speaking, field margins can be an effective measure reducing exposure of NTAs to pesticides and limiting the run-off of pesticides into the water (Burn, 2003; Felsot et al., 2011; Holland et al., 2020; Marshall and Moonen, 2002; Reichenberger et al., 2007) but only if sufficient width and density of the margins are assured. Our findings support the importance of field margins as a supporting mitigation measure, as they substantially magnified effects of reduction of insecticide-driven lethality rate from LR90 to LR70, resulting in positive impacts on beetle populations comparable with those achieved with reduction of lethality rate alone to LR50. Therefore, adding sufficiently broad off-field habitats should help to maintain viable beetle populations in agricultural landscapes even with moderate use of insecticides.

Adding / increasing field margins in agricultural landscapes would benefit not only ground beetles, but many other arthropods including staphylinid species, and spiders, for which semi-natural habitats adjacent to fields can act as reservoirs (Pfiffner and Luka, 2000). For some groups of arthropods the observed effects of increasing field margins could be even more pronounced. For example, survival of pollinators with herbivorous larval stages depend on habitats providing host plants, and properly managed field margins can fulfill this role (Thomas et al., 2000). It is, however, important to note that different types of field margins (grassy margins, hedgerows, flower strips) support at best different groups of arthropods (Pywell et al., 2005), and that needs to be taken into account when planning management strategies in agroecosystems.

4.2. Simulated winter survival of beetles versus population declines under scenarios with high and moderate insecticide-driven lethality rate

Winter mortality is a key factor influencing arthropod survival in temperate climates (Knapp and Saska, 2012). Although the *Bembidion* model applied in this study used soil temperatures at depth of 5 cm instead of air temperatures to calculate overwintering mortalities according to the equations by Petersen et al. (1996), this still resulted in quite low overwintering survival, with mean for the years 2010–2019 of 46.2%, dropping to only 10% during harsh winters. The survival rates of *B. lampros* over winter observed in our simulations were lower than those reported by Knapp and Saska (2012) (80% on average) or Petersen et al. (1996) based on which the *Bembidion* model was initially calibrated for the Danish agricultural landscapes (Bilde and Topping, 2004). Therefore, in our simulations beetle populations in some years were heavily reduced due to high overwintering mortalities, which in combination with mortality due to exposure to insecticides in the crop habitats resulted in long-term decreasing population trends in scenarios with high and moderate insecticide-driven lethality rate. Only in scenarios with the highest reduction of insecticide-driven lethality (LR10) and in scenario with no pesticide-related lethality (not shown in this study), the modeled beetle populations were stable over the simulation period of 40 years.

We suggest two alternative explanations of this phenomenon. First, the overwintering mortality rates from our simulations are not overestimated and are indeed experienced by beetles in field conditions. Although that would need confirmation from further field studies, it is probable especially that, as stated by Knapp and Saska (2012), the use of impermeable isolators in their study provided partial protection against low temperatures. That would suggest that high overwintering survival rates reported by Knapp and Saska (2012) and Petersen et al. (1996) (in both studies isolators were applied) could be overestimated compared to real-field conditions outside the isolators. The second possible explanation assumes that the overwintering mortalities predicted in our simulations are overestimated. This could happen if beetles living in climatic conditions similar to those encountered in Poland have some adaptive mechanisms allowing them to cope with low temperatures better, or more probably, soil temperatures at the 5 cm depth do not represent well the real thermal conditions experienced by beetles in overwintering habitats. Indeed, the soil temperatures we used were measured under the bare ground, while beetles rather prefer grassy overwintering habitats (Petersen, 1999) that can further buffer temperature fluctuations, especially when covered with a snow layer. In addition, Dennis et al. (1994) reported the presence of *Bembidion* species in soil samples taken from field boundaries to a depth of up to 35 cm, which suggests that, if needed, beetles can hide in deeper layers of soil to avoid low temperatures.

4.3. Importance of landscape structure and management implications

The use of nine study areas differing in both landscape and farmland heterogeneity allowed us to investigate the impacts of landscape structure on population dynamics of beetles. We found that both mean overall beetle density and mean occupancy of the modeled *B. lampros* populations were positively influenced by landscape diversity. Our findings therefore confirm the importance of the landscape heterogeneity in supporting carabids (Bertrand et al., 2016; Vanbergen et al., 2005). At the same time, the beetle populations depended strongly on the amount of semi-natural habitats in the landscape. LU and DO, being study areas with the lowest share of semi-natural herbaceous habitats (~ 5%), hosted more than four times lower beetle populations which occupied two times smaller area than in study areas SK and WA where such habitats represent 16–20% of the area. It is therefore of utmost importance to protect and maintain semi-natural grassy habitats as they play a key role in maintaining biodiversity in agricultural landscapes (Arponen et al., 2013; Harlio et al., 2019), especially that the drastic loss of those habitats has been observed in the last decades (Cousins et al., 2015; Stoate et al., 2009).

Our findings also clearly show that the effectiveness of mitigation measures applied strongly depends on landscape heterogeneity, as we found the highest impacts of the measures tested in study areas with the lowest landscape diversity and proportion of semi-natural habitats. For effectiveness of off-crop measures the farmland structure was also important, i.e., effectiveness was higher in study areas with high aggregation of patches, dominated by large fields, and with low density of field boundaries. Therefore, reaching the same management goal, e.g., increasing the overall beetle density in a landscape by 20%, would require different strategies depending on the landscape structure. In case of our study areas, assuming long persistence of the insecticides in the environment (DT_{50} of 25 days), this goal could be reached by decreasing of insecticide-driven lethality rate up to LR70 in DO study area, to LR50 in study areas NG, KR, LU, WI, LI, to LR30 in study areas RE and SK, and to LR10 in study area WA. Applying insecticides characterized by lower DT_{50} would reach the same goal with correspondingly higher LRs. Similarly, the effects of adding margins of the same width to the same proportion of agricultural fields results in different increase in field margins abundance in a landscape depending on the farmland heterogeneity, and this, in turn, will modify the dynamic of beetle populations.

Depending on landscape and farmland heterogeneity, certain combination of mitigation measures may be not feasible due to high cost of application. For example, increasing the abundance of field margins is always associated with reduction of cropland area (and potentially reduced yields), increased workload and costs of field margins management (costs of mowing, mulching and re-sowing). In landscapes with predominance of small fields, the overall costs of introduction of even narrow field margins could be too high for some farmers. For instance, in some parts of Poland fields are so narrow that introducing even 2 m wide field margins would be problematic, therefore, 4 m margins which in some cases were the margin mitigation allowing for a significant improvement of *B. lampros* population dynamics, would be impossible. In such cases, the reduction insecticide-driven lethality is more feasible mitigation option.

5. Conclusions and recommendations

The study showed that reducing the insecticide-driven lethality has generally a bigger positive impact on beetle density and occupancy than increasing the abundance of field margins in a landscape. However, field margins can serve as a key supporting mitigation measure, as they can substantially magnify effects of reducing insecticide-driven lethality, providing important refuge and overwintering habitats for beetles. We demonstrated that sufficiently broad off-field habitats can help maintaining viable NTA populations in intensively managed agricultural landscapes even with moderate use of insecticides. This is an important finding since substantial reduction of the pesticide use in the near future seems highly improbable.

By analyzing beetle population dynamics along a gradient of landscape heterogeneity we showed the importance of landscape context when selecting the best mitigation measures, meaning that reaching the same management goal may require different strategies depending on the landscape structure. At the same time different measures are to varying degrees possible to implement in differently managed landscapes, e.g., very wide field margins are impossible to apply in fragmented landscapes with many small fields. The importance of environmental conditions and landscape structure in population modelling for risk assessment, and the need for a more holistic (systems) view have been earlier stressed by Schäfer et al. (2019), Streissl et al. (2018) and Topping et al. (2015). As shown by our study, individual-based population modelling at a landscape level, using ALMaSS or similar tools, is a promising method that can be suited for any specific landscape and allows for testing multiple scenarios employing different combinations of a range of mitigation measures. The modelling, when combined with cost-benefit analysis, could become an important tool in decision making and environmental risk assessment (Topping et al., 2019).

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CRediT authorship contribution statement

Elżbieta Ziółkowska: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Visualization, Writing - original draft. **Chris J. Topping:** Conceptualization, Methodology, Software, Data curation, Writing - review & editing. **Agnieszka J. Bednarska:** Conceptualization, Methodology, Writing - review & editing. **Ryszard Laskowski:** Conceptualization, Methodology, Writing - review & editing, Project leadership, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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